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Effects of landslides on contaminant sources and transport in steep pastoral hill country

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Abstract

The effects of landslides on contaminant sources and transport during storms in steep pastoral hill country were evaluated for a small, first-order hillslope in the Lake Tutira catchment, New Zealand. Nitrate (NO_3^--N) concentrations were significantly higher in soil water than in subsurface water in the saturated zone, in surface runoff, and in a stream draining the hillslope. Concentrations in colluvium and undisturbed areas were higher than in landslide debris, scars and the stream. Stream NO_3^--N was lower due to in-stream processes, and was derived primarily from soil water in colluvium and undisturbed areas where livestock congregate and waste accumulates. Concentrations of dissolved reactive phosphorus were higher in streams than in other types of water, particularly greater than in runoff from scars. Colluvium and other unmonitored source areas were contributing dissolved reactive phosphorus to the stream. Landslide scars and debris were not significant sources of NO_3^--N or dissolved reactive phosphorus because of the lack of soil, vegetation, livestock and waste in these areas. Soil and subsurface water had the highest levels of total dissolved solids because of the relatively long residence times and leaching of solutes. Landslide-disturbed areas had the greatest dissolved solids concentrations and, with the exception of nutrients, were sources of solutes transported to the stream via subsurface pathways.

During a storm NO_3^--N loadings were largest from colluvium and debris subsurface discharges. Loadings of total P in runoff were much higher than those of dissolved reactive phosphorus and were greatest from debris. They were primarily composed of particulate P and were a significant source of elevated stream phosphorus values. Loadings of total dissolved solids in runoff were also highest from debris, and loadings in subsurface water were greatest from debris and colluvium.

Landslides can alter the sources and transport of contaminants in steep, agricultural hill country directly through physical disturbances that affect

pollutant generation and movement, and indirectly by altering livestock use of areas. These effects should be considered when modelling contaminant transport and managing these types of catchments.

Introduction

Sheep and cattle grazing can be a major cause of non-point source pollution of ground and surface water in New Zealand. Contaminants can include sediment derived from erosion, nutrients such as nitrogen (N) and phosphorus (P), and pathogens. High levels of orthophosphate, or dissolved reactive P (DRP), and nitrate (NO_3^--N) can cause eutrophication of warm, slow-moving water. The impacts of grazing have been well documented in New Zealand and other countries, and are often accentuated in catchments that are unsuitable or only marginal for pastoral uses (Mosley and Rowe, 1981; Cooke and Dons, 1988; Cooper and Thomsen, 1988). Much of the hill country on New Zealand's North Island is particularly sensitive because of deforestation, very steep terrain and thin, unstable soils (Crozier *et al.*, 1980; Merz and Mosley, 1998). This area is prone to extensive landsliding during frequent high intensity winter rains and occasional cyclonic storms. Cyclone Bola in 1988, for example, caused severe landsliding in the hill country, particularly in Gisborne and northern Hawke's Bay. Grazing compounds the impact of erosion and sedimentation by introducing nutrients to streams and lakes. For example, Lake Tutira, a small lake located 35 km north of Napier, is recognised for its ecological and amenity values, but has experienced severe eutrophication (McColl, 1978). The lake has a steep catchment used primarily for grazing and was severely affected by landslides during Cyclone Bola and at other times in the past (Page *et al.*, 1994; Merz and Mosley, 1998).

Although the hydrology and nutrient dynamics of forested and pastoral catchments have been extensively studied, particularly in New Zealand (Mosley, 1979; Mosley and Rowe, 1981; Pearce *et al.*, 1986; Cooke and Dons, 1988; Cooper and Thomsen, 1988), very little is known about how landslides affect these processes in steep pastoral catchments. Landslides can potentially affect the sources and movement of water and nutrients because they modify the topography of hillslopes and the physical characteristics of soils (Fig. 1). Soil or regolith depth is reduced considerably on fresh landslide source areas, exposing a lower soil horizon or the underlying bedrock. The loss of most soil and vegetation on the scars dramatically reduces infiltration and water-holding capacities (Merz and Mosley, 1998). Both the colluvial and unweathered material in fresh scars is denser than in undisturbed soils (Preston, 1996). Deposits or debris from the source area accumulate downslope, increasing soil depth, breaking down

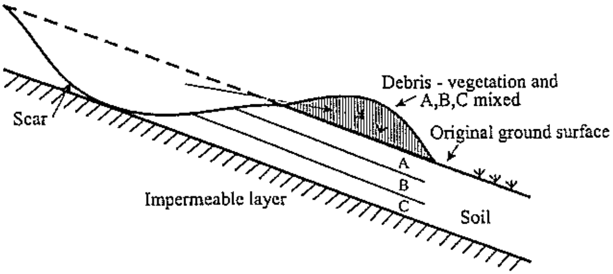

Figure 1 – Schematic cross-section of rotational landslide with scar and debris.

soil structure, and bringing lower soil horizons from the source area to the surface (Fig. 1). This results in porosities, water-holding capacities and drainage that are less than in undisturbed areas. Pores can also become blocked with fine particles, and vegetation can be buried or removed during the soil movement and deposition (Jensen, 1998). In older debris, however, many of these physical soil characteristics recover over time and may become greater than in undisturbed locations, where there is compaction by livestock trampling (Merz and Mosley, 1998).

Runoff generation in an area subject to landsliding appears to be controlled more by the presence of saturated areas in valley bottoms and swales than by the extent or characteristics of landslides (Merz and Mosley, 1998). Localised increases in surface runoff from landslide scars, however, can erode the remaining soil and the deposits downslope (Merz and Mosley, 1998). Water in, or draining from, soils disturbed by landslides can also have chemical characteristics that differ from water in undisturbed soil because the water generally has a shorter residence time due to decreased porosity and water–holding capacity. In addition, lower horizons are typically exposed in the disturbed areas and can influence the chemical composition.

The objective of this study is to evaluate whether landslides alter the sources and transport of contaminants in steep, agricultural hill country. The hypothesis is that contaminant concentrations and loadings in water within, or draining from, locations and soils disturbed by landslides are different from those in undisturbed locations. NO_3^--N and dissolved reactive phosphorus (DRP) are examined in detail because they are common agricultural pollutants that can cause eutrophication of downstream water bodies. Total dissolved solids are also considered because it is an indicator of all solutes, including those derived from different soil horizons, and of subsurface residence times and hydrologic pathways.

Study area

The study area is a small (1.5 ha), first-order catchment on the southern boundary of the Lake Tutira watershed (Fig. 2) and lies approximately 1 km east of the lake. It ranges in elevation from 280 to 370 m asl., and has a northwest aspect. The catchment is typical of steep, erosion- and landslide-prone hill country of the eastern North Island. The catchment is underlain by marine sandstone, siltstone, and mudstone, interbedded with bands of coarse limestone, dipping 2-10° to the southeast (Merz and Mosley, 1998; Jensen, 1998). Soils are primarily intergrades between Yellow-brown Pumice soils and Yellow-brown Loams, and tephra is widespread in the area (Eden *et al.*, 1993; Page *et al.*, 1994). The regolith is naturally prone to instability due to its shallow depth and high permeability above the bedrock, and perched water tables are common at the bedrock surface. The study area has been disturbed by landslides over about 70% of the catchment, and pipe and gully erosion is also present.

The catchment is primarily pasture composed of ryegrass, white-clover, brown top, and rats tails, with some kanuka-manuka-gorse shrubland along the ephemeral stream channel. Sheep and cattle graze throughout the catchment, and a farm track divides the catchment into an upper and lower area. The original forest cover has been removed since the arrival of the Polynesians, and pasture was developed during European settlement. Forest removal has accentuated landsliding (Trustrum and Page, 1992).

The mean annual precipitation (1951-80) at Tutira, 2.5 km west of the site, is 1440 mm. Thirty-two percent of the annual rainfall occurs during the wettest months of July-August, and 19% during the driest months of September-November. Precipitation varies both seasonally and from year to year, with prolonged summer droughts and intense autumn rains. The study period during winter 1997 and autumn 1998 was drier than normal.

Data collection

The study area was divided into different contemporary land-surface condition (CLC) classes (modified from Preston, 1996): undisturbed, landslide debris, well-developed colluvium, and scars (Fig. 2). Preston (1996) developed this classification system for areas in the Lake Tutira catchment, based on erosion by landslides and differences in geomechanical properties. The system can be used to classify landslide erosion in the catchment and evaluate potential effects on water and contaminant sources and movement.

Based on the modified CLC classes eight sites on the hillslope were selected to measure discharge and water chemistry: two in undisturbed areas, two in debris, two in colluvium, and two at scars (Fig. 2, Table 1). Discharge

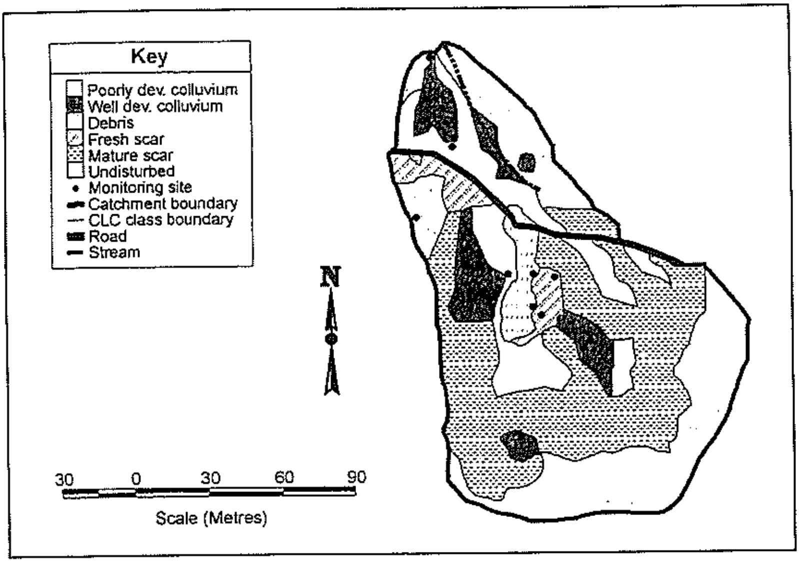

Figure 2 – Areas within contemporary land-surface condition (CLC) classes and monitoring sites in the study catchment.

Table 1 – Monitoring sites, contemporary land-surface condition (CLC) class, types of water and other characteristics.

Site/CLC Class	Type of Water	Distance from weir (m)	Mean slope angle (deg.)	Drainage area (m^2)
Scar	surface	116	43	319
Scar	surface	133	54	64
Undisturbed	surface soil subsurface	8	40	50
Undisturbed	surface soil subsurface	81	30	77
Colluvium	surface soil subsurface	50	38	49
Colluvium	surface soil subsurface	107	43	111
Debris	surface soil subsurface	109	29	271
Debris	surface soil subsurface	123	44	228
Pipe	subsurface	109	44	47

and water chemistry were also measured in one subsurface pipe, and in the stream to which the entire hillslope drains. Precipitation was measured near one of the undisturbed sites and the stream monitoring site, using a 0.2 mm tipping-bucket raingauge connected to a Campbell RRDL-3 data logger.

Surface runoff

Surface runoff collectors were constructed at each hillslope site except at the pipe (Fig. 3). For all sites except the scars, the collectors consisted of a strip of thin sheet metal approximately 300 cm long x 30 cm wide placed horizontally on the ground surface perpendicular to the slope. The sheet metal was inserted approximately 5 cm laterally into, and 3 cm under, the grass and root layer on the upslope side, and fed into a polyvinyl chloride (PVC) rain gutter leading to a tipping bucket (1-litre compartments) for flow measurements (Woods and Rowe, 1996).

To measure overland flow from the two landslide scars, collection troughs 5 cm deep were excavated immediately downslope. They were sealed with a thin layer of concrete, and runoff was led to weir boxes constructed of treated plywood, with volumes of 480 L. Weir boxes were used because it was expected that more discharge would be generated from the scars than from other areas, and that tipping buckets would be inadequate for measuring the flows. The weir boxes had 28° 04' V-notches with a sharp crest (thickness of 1 mm), and pressure transducers attached to a CR10 datalogger continuously measured stage. The transducers were calibrated to the discharge from the weirs, and the KindsvaterShen equation (ISO 1993) was used to estimate discharge.

Subsurface discharge

At all sites except the scars and pipe, subsurface flow in the saturated zone was collected and measured using shallow trenches dug perpendicular to the hillslope immediately downslope from the surface collectors (Fig. 3). Each trench was 1 to 2 m long and excavated to a depth of 1 to 2 m, where a relatively impermeable clay layer was reached. The bottom of the trench was sloped slightly to one end and sealed with a thin layer of concrete, and runoff was led to a tipping bucket. The trenches were covered with tarpaulins to exclude precipitation. The upslope surface runoff collectors minimised overland flow to the trenches.

Pipe flow

The outflow point of the subsurface pipe was excavated slightly as a small pit, the bottom sealed with a thin layer of concrete, and a funnel enplaced at the downslope end of the pit. The funnel led via plastic tubing to a weir box.

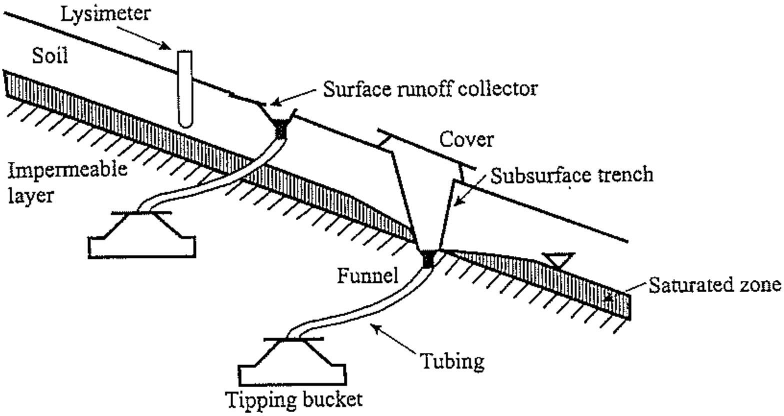

Figure 3 – Schematic diagram of surface runoff collector and subsurface trench.

Stream discharge

A 90° V-notch weir was used to measure stage in the main stream draining the hillslope (Merz and Mosley, 1998). The weir was adequate for measuring flows from the catchment of up to 37 L/s. A pressure transducer and RRDL-3 datalogger were used to measure and record stage behind the weir.

Water chemistry

Water samples were collected from each site during several precipitation events. Three were during winter (two small events on 17 August and 6 September 1997, and one larger event on 21-22 August 1997) and one was in autumn (a large event on 9 April 1998). 'Catching' storms was a difficult task, given their unpredictability and the remote location of the study area. In addition, the 1997-98 period was unusually dry in Hawkes Bay.

Grab samples were collected from the outflow of the tipping buckets and weir boxes during the precipitation events. Not every location had adequate flow during these events, and difficulties were encountered collecting samples from every site under storm conditions. However, samples were collected from most of the sites, providing data from representative locations and CLC classes.

An Isco Model 3700 automatic sampler was used at the weir in the stream to collect sequential samples of discharge at 15-minute intervals during the storm on 21 August 1997. This was the only storm that generated enough water in the stream to be sampled, due to the unusually dry conditions during the investigation period. An automatic actuator for the sampler was secured on the inside of the weir 5 cm above the V-notch for a discharge of approximately 0.27 L/s. The average of the peak storm flows measured previously by Merz (1997) was approximately 2 L/s. Precipitation events were 5 hours in duration on average, and stream discharge lasted an average of 15 hours.

In addition to grab samples from the stream, surface collectors, and subsurface trenches, vadose zone water samples were collected immediately before and after precipitation events using suction cup lysimeters. These were installed immediately upslope of each subsurface trench, at a depth approximately 10 cm above the impermeable layer. At one of the debris sites with a thicker layer of debris, two lysimeters were installed: one at 10 cm above the impermeable layer, and one at approximately 50 cm below the ground surface. The lysimeters were polyvinyl chloride tubes with an outside diameter of 1.9 inches and porous ceramic cups at the bottom end. To sample from the lysimeters a vacuum of 70 centibars was created in the tube, and water held in the soil at tensions of less than 70 centibars was sucked into the lysimeters through the porous cups over several hours, depending on the soil water content. After the required amount of time the water was pumped out of the lysimeter into a sample bottle.

One-litre plastic bottles were used for the automatic sampler. Each of these samples, and the samples from each surface runoff collector, subsurface trench, and vadose zone site, were split into two sub-samples. Each sub-sample was put in a 600-mL plastic container. One sub-sample from each site was cooled to 4°C with ice and sent to R.J. Hills Laboratory in Hamilton for total P analysis using nitric/perchloric acid digestion and Molybdenum Blue colorimetry. The other sub-sample was analysed for DRP and NO_3^--N using a Hach DR/2000 Spectrophotometer. DRP was analysed using the PhosVer3 (asorbic acid) Method. NO_3^--N was analysed using the Cadmium Reduction Method. All samples were analysed within 24 hours of collection. Temperature and conductivity were measured in the field using a portable Hach Conductivity meter. Total dissolved solids were estimated as conductivity multiplied by 0.6. Because many other cations and anions were not analysed, ionic balances were not computed to compare to the estimated dissolved solids values.

Data analysis

NO_3^--N and DRP values in the stream were compared to water quality criteria/ guidelines for DRP and dissolved inorganic nitrogen for lakes and streams cited by Smith *et al.* (1993). These upper limits are both a 'concentration above which water may be degraded with regard to the specified use.' The value for DRP is 0.01 mg L^{-1}, and for dissolved inorganic nitrogen is

0.1 mg L^{-1}; concentrations above these values can cause undesirable profuse biological growths impairing contact recreation and aesthetics.

Concentration data for each analyte were grouped by type of water: stream water, surface runoff, soil water (vadose zone), and subsurface water (saturated zone). Data were also grouped by CLC class along with the stream. Boxplots were developed for each analyte for these groups, and were used to visually assess and compare the data. Because many of the data were not normally distributed, non-parametric Kruskal-Wallis tests were used (95% confidence level) to evaluate differences.

In addition, equivalence tests were used, based on a difference of ≥20% between groups, and a 95% confidence level. These tests have been used widely in the drugs testing industry and have only recently been recommended for use in environmental science and management. They have considerable advantages over the use of traditional null hypothesis tests (such as *t*-tests or ANOVA) (McBride *et al.*, 1993; McBride, 1999). A difference of 20% (calculated as the absolute difference in means between two stations divided by the larger mean) or more was considered to be important for targeting significant contaminant source areas. However, differences ≥10% were also evaluated and a 90% confidence level was used to assess the sensitivity of these tests to these parameters.

Concentrations of the analytes in different types of water (at different depths) were also plotted to evaluate changes and differences with depth during the storms of 17 August 1997 and 6 September 1997 at the debris site where two lysimeters were used.

The loads per unit time (mg min^{-1}) and for the entire storm (mg) for each analyte were estimated for surface runoff from colluvium, debris, and undisturbed material, as well as for subsurface flows from colluvium and debris during the storm on 22 August 1997. This was the only storm, and these were the only locations, for which discharge and concentration data were collected that were adequate to estimate loads. The mean concentration (mg L^{-1}) for the event was multiplied by the mean discharge (L s^{-1}) to estimate instantaneous loads (mg s^{-1}, which was then converted to mg min^{-1}), and by total flow volume (L) to estimate the total storm loads (mg), for each location. Specific loads (mg m^{-2} s^{-1}) were also computed by dividing the load by the drainage area for each site, which was determined from a detailed topographic map. Loads were then compared among locations.

Results and discussion

Concentrations

NO_3^--N

NO_3^--N was not detected in precipitation during the one storm sampled on 21 August 1997 (Table 2). The highest values in surface runoff (up to

3 mg L^{-1}) from all areas occurred during the 4 September storm. One high value (1.2 mg L^{-1}) occurred in pipe flow during the 21 August storm. Several very high values (>7 mg L^{-1}) were measured in debris soil water, both near the surface and at depth, on 17 August and 6 September. During the storm of 22 August, NO_3^--N was not detected in the stream until more than three hours after flows began that were high enough to trigger the automatic sampler (Fig. 4a). The NO_3^--N concentration was highest five hours after the start of stream flow (0.9 mg L^{-1}).

Table 2 – Analyte concentration (mg L^{-1}) summary statistics by type of water

Type of Water	NO_3-N	DRP	TP	TDS
Precip.				
N	1	1	1	1
Value	0.00	0.030	0.93	40
Surface				
N	19	18	14	19
Mean	0.60	0.174	0.79	124
Median	0.50	0.000	0.18	108
Max	3.00	0.870	5.40	412
Min	0.00	0.000	0.04	1
St Dev	0.70	0.292	1.43	121
Soil				
N	14	14	7	14
Mean	3.73	0.106	1.07	203
Median	1.80	0.020	1.00	248
Max	16.00	0.710	2.40	320
Min	0.00	0.000	0.20	40
St Dev	4.80	0.206	0.69	105
Subsurface				
N	9	9	-	9
Mean	0.41	0.040	-	226
Median	0.40	0.000	-	259
Max	1.10	0.350	-	280
Min	0.00	0.000	-	10
St Dev	0.33	0.116	-	84
Stream				
N	8	8	-	8
Mean	0.24	0.246	-	117
Median	0.00	0.255	-	117
Max	0.90	0.370	-	133
Min	0.00	0.140	-	98
St Dev	0.35	0.081	-	12

Note: - not analysed DRP - dissolved reactive P
TP - total P TDS - total dissolved solids

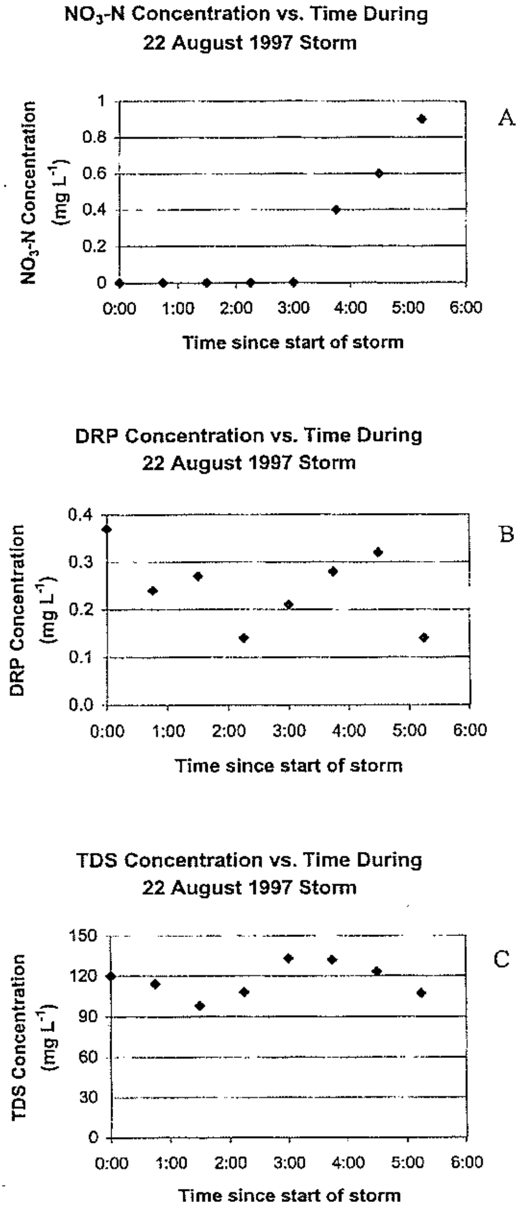

Figure 4 – Concentrations versus time in stream during storm of 22 August 1997 for (A) NO_3^--N, (B) DRP and (C) total dissolved solids (TDS).

The Kruskal-Wallis tests showed that soil water had a significantly higher ($P<0.05$) median NO_3^--N concentration than the stream, surface runoff and subsurface water (Table 2, boxplots in Figure 5a). Subsurface water and surface runoff also had higher values than the stream. The equivalence tests showed that the mean soil water concentration was not equivalent to (was $\geq 20\%$ more than) values for other types of water ($P<0.05$). No other significant differences $\geq 10\%$ or for $P<0.1$ were found.

In thick debris, NO_3^--N concentrations were considerably higher in shallow soil water (>15 mg L^{-1}) than in surface runoff or at greater depths (<1 mg L^{-1}) during the first storm (Fig. 6a). For the second event, NO_3^--N concentrations increased with depth to a maximum in deeper soil water (>12 mg L^{-1}), but were still <1 mg L^{-1} in the saturated zone.

These results show that NO_3^--N levels in soil water were significantly higher than in the other types of water. Values in surface runoff and subsurface water were also somewhat higher than in stream water. This suggests that soil water is a reservoir and significant source of NO_3^--N to the stream. Fertilisers, animal waste, natural N in soils, and fixation by plants such as clover are all sources of soil NO_3^--N. This is an inorganic form of N that can cause eutrophication in conjunction with P (Burt *et al.*, 1993). Animal waste can also be a significant source of organic N, including ammonia, which is common in agricultural environments. Ammonia is toxic to many species of fish at very low concentrations. Some organic N is also oxidised to form NO_3^--N through nitrification. Transport of N through the hillslope to the stream is predominantly as dissolved NO_3^--N and can include overland flow during storms as well as saturated subsurface flow (Burt *et al.*, 1993). N concentrations in soil water and, ultimately, in stream water are affected by these transport pathways and the residence time of water, during which it is in contact with soil particles and contaminant sources. NO_3^--N can also be diluted in surface runoff and lost during transport through uptake by vegetation and microbes and by denitrification, thereby contributing to the lower concentrations in the stream (Cooper and Cooke, 1984; Goulding *et al.*, 1996). Concentrations can also decrease in the stream through dilution by stream water, uptake by aquatic biota, denitrification, and rapid transport downstream as a highly soluble and mobile contaminant (Cooper and Cooke, 1984; Burt *et al.*, 1993). These findings are similar to those for agricultural and other areas unaffected by landslides, particularly with regard to high soil water N levels that are a source of N in the near-stream zone (Pionke *et al.*, 1988; Goulding *et al.*, 1996; Ohrui and Mitchell, 1998).

The undisturbed and colluvium sites had higher ($P<0.05$) median NO_3^--N concentrations than debris, and debris had a higher value than the stream and scar (Table 3, Fig. 5b). Mean concentrations in the undisturbed area

Table 3 – Analyte concentration (mg L^{-1}) summary statistics by contemporary land-surface condition (CLC) class.

CLC Class	NO_3-N	DRP	TP	TDS
Scar				
N	6	6	4	6
Mean	0.58	0.008	1.79	237
Median	0.05	0.000	0.83	220
Max	3.00	0.040	5.40	412
Min	0.00	0.000	0.11	108
St Dev	1.19	0.016	2.49	115
Debris				
N	19	19	6	19
Mean	2.40	0.126	0.91	216
Median	0.50	0.000	0.75	249
Max	16.00	0.730	2.40	320
Min	0.00	0.000	0.09	36
St Dev	4.50	0.238	0.88	83
Colluvium				
N	10	10	6	10
Mean	1.06	0.146	0.49	121
Median	0.89	0.015	0.50	120
Max	2.60	0.670	0.93	250
Min	0.30	0.000	0.06	1
St Dev	0.68	0.238	0.39	117
Undisturbed				
N	7	6	5	7
Mean	1.09	0.178	0.60	71
Median	1.10	0.035	0.20	40
Max	2.20	0.870	1.63	320
Min	0.00	0.000	0.04	1
St Dev	0.88	0.342	1	111
Stream				
N	8	8	-	8
Mean	0.24	0.246	-	117
Median	0.00	0.255	-	117
Max	0.90	0.370	-	133
Min	0.00	0.140	-	98
St Dev	0.35	0.081	-	12
Pipe*				
N	3	-	-	3
Mean	0.60	-	-	232
Median	0.60	-	-	267
Max	1.20	-	-	284
Min	0.00	-	-	146
St Dev	0.60	-	-	75

Notes: - not analysed
DRP - dissolved reactive P
TDS - total dissolved solids
* not used in hypothesis or equivalence testing
TP - total P

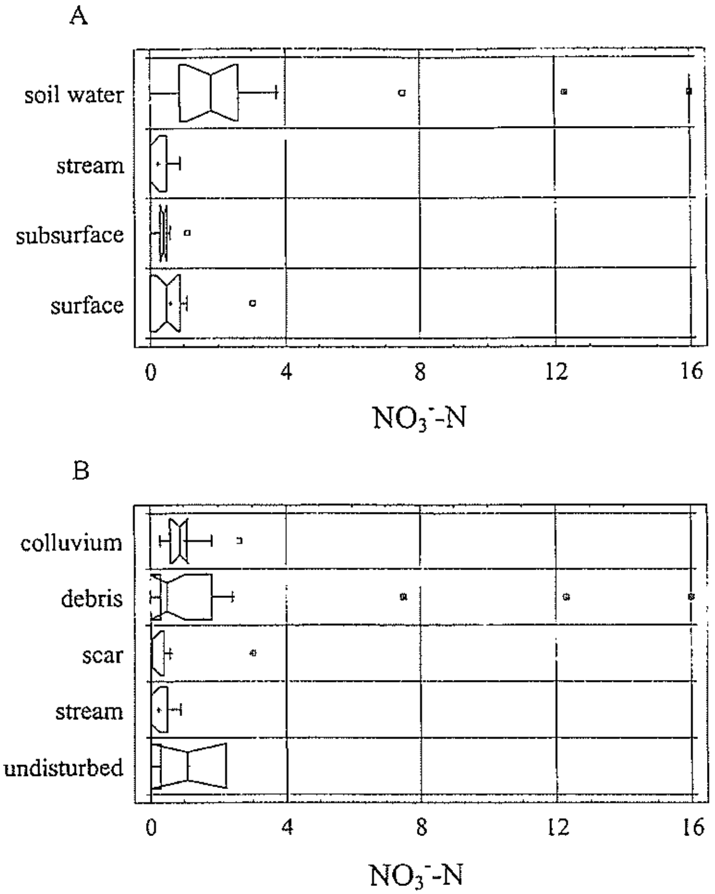

Figure 5 – Boxplots of NO_3^--N concentrations (mg L^{-1}) by (A) types of water and (B) CLC class. Boxes show the 25th percentile, median, and 75th percentile. The mean is a cross near the middle. Whiskers extend from the end of the box to the smallest point within 1.5 interquartile ranges from the 25th percentile, and to the largest point within 1.5 interquartile ranges from the 75th percentile. Values within 3 interquartile ranges are plotted as points, and far outside values are plotted as points with a cross in the middle. Notches show an approximate 95% confidence interval for the median.

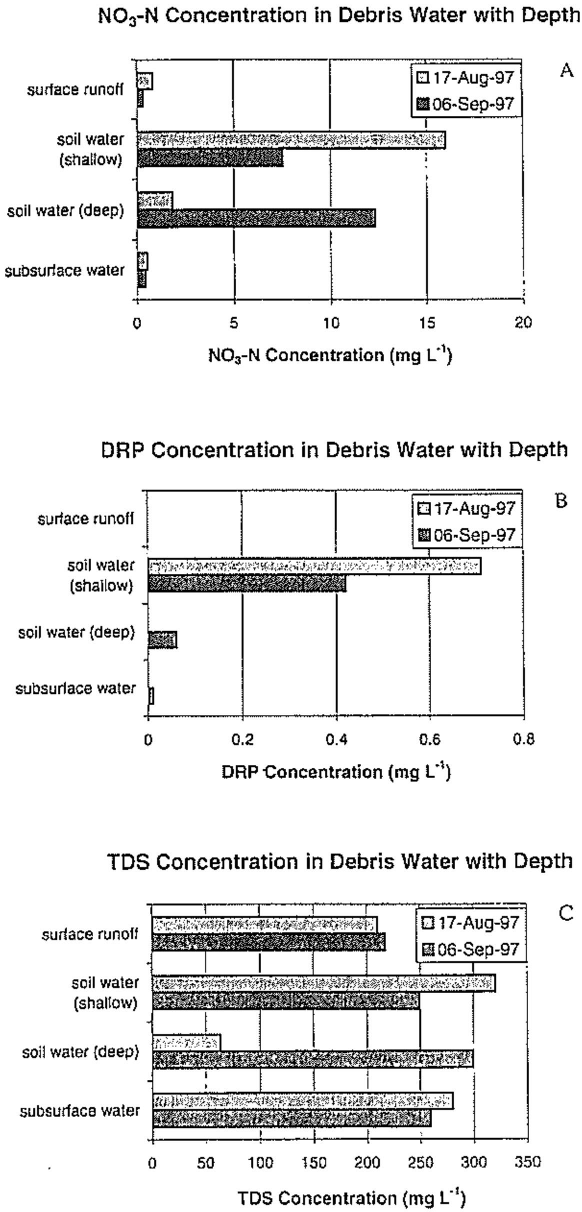

Figure 6 – Concentrations in debris water with depth for (A) NO_3^--N, (B) DRP and (C) total dissolved solids.

and colluvium were ≥20% more than values in the stream ($P<0.05$). The value in the debris was also ≥20% higher than in the stream for $P<0.1$.

These results indicate that, in general, NO_3^--N levels in undisturbed areas and colluvium were considerably higher than in the stream and scars. They were also somewhat higher than in debris, and values in debris were slightly higher than in the stream and scars. Scars do not have enough soil cover or water-holding capacity to be a source of NO_3^--N in surface runoff, when compared to other areas with more soil. Livestock also generally avoid scars, and debris to a certain extent, because of the steep slopes and lack of grass. This corresponds to the findings of Merz and Mosley (1998), who observed that stock tend to graze, and compact the soil by trampling, in undisturbed areas on spurs rather than on the midslopes where many landslides occur. This results in less animal waste and lower NO_3^--N values in disturbed areas than in colluvium or undisturbed areas. NO_3^--N in debris can also be leached faster due to the mixed, unstructured nature of the soil. N from animal waste, in addition to N from fertiliser and fixation, can accumulate in soil in colluvium and undisturbed areas grazed by livestock. These results suggest that NO_3^--N is derived primarily from colluvium and undisturbed areas and transported via subsurface pathways to the stream.

Dissolved Reactive Phosphorus

The DRP concentration in precipitation during the storm of 21 August 1997 was very low (0.03 mg L^{-1}, Table 2). All of the elevated values in surface runoff occurred during the storm of 9 April 1998, as with NO_3^--N. The highest value was from the undisturbed area (0.87 mg L^{-1}); concentrations in runoff from the scar were much lower. In debris soil water, two values (on 17 August and 6 September) were >0.4 mg L^{-1} in the shallow soil zone. There was only one high concentration in subsurface water in colluvium (0.35 mg L^{-1}). DRP was not detected in the two pipe flow samples, indicating that there were no significant sources of P contributing to the pipes and P was diluted, given the high water discharges in the pipes during storms. In stream water, concentrations were consistently higher (up to 0.37 mg L^{-1}) than values at most other locations during the storm sampled in August (Fig. 4b). However, maximum values in the stream during the August storm were lower, and values were less variable, than in soil water, surface runoff, colluvium and debris during other storms.

The median stream DRP concentration was higher ($P<0.05$) than that in surface runoff and soil water, and they both had higher values than subsurface water (Table 2, Fig. 7a). The mean stream concentration was ≥20% more than the value in subsurface water ($P<0.05$). However, for $P<0.1$ and when the difference in values was reduced to ≥10%, the stream value was also greater than values in surface runoff and soil water.

In thick debris, DRP was not detected in surface runoff during the storms of 17 August and 6 September, but increased considerably and was at maximum concentration in shallow soil water (Fig. 6b). During the first storm, DRP was not detected in soil water at greater depths or in saturated subsurface water. During the second storm, DRP was detected at greater depths but decreased considerably with depth in soil water, to <0.02 mg L^{-1} in subsurface water. P is not transported very far down into the soil profile during or immediately after storms. This suggests that P is either more likely to be transported laterally to the stream in shallow subsurface water, or stored in the upper soil. This pattern is consistent with many other studies that have found that most P is stored in upper soil layers (Sharpley *et al.*, 1994; Daniel *et al.*, 1994).

These results demonstrate that DRP concentrations in stream water were significantly higher than in subsurface water, soil water and surface runoff in the areas sampled. This suggests that P is generated from source areas and transported to the stream via flow paths that were not represented by our sampling sites. Sources of P include fertilisers (particularly superphosphate), animal waste, and natural P in some soils (Daniel *et al.*, 1994), which are all problems in the Lake Tutira catchment (McColl, 1978). As with N, animal waste can accumulate and be a P source in areas where livestock congregate. Flowpaths can include overland flow and saturated subsurface flow. Although DRP is the predominant dissolved form in water, P is highly bound to sediment and a large portion of P transport can be associated with erosion and movement of sediment in overland flow (Pionke *et al.*, 1988; Ohrui and Mitchell, 1998). Therefore, particulate P which we did not sample transported to the stream, possibly from enhanced erosion of disturbed areas, can also contribute to elevated stream DRP levels when P redissolves in the water column (McColl, 1978; Sharpley *et al.*, 1994; Daniel *et al.*, 1994; Merz and Mosley, 1998). Some mineral P can also be lost during transport through biotic uptake, and fixation or adsorption of P to immobile soils (Cooper and Cooke, 1984; Goulding *et al.*, 1996).

The stream also had a higher median DRP value than colluvium and undisturbed areas, and these areas had higher values than debris and the scar (Table 3, Fig. 7b). The mean stream concentration was ≥20% more than in the scar ($P<0.05$). For $P<0.1$, however, the value in colluvium was also ≥20% more than that in the scar.

Stream DRP concentrations were significantly higher than values in water from other CLC classes, particularly in the scar. Values in colluvium and undisturbed areas were also somewhat higher than in disturbed areas, indicating that disturbed areas do not contribute much DRP to the stream. As with N, this results from the lack of livestock in these areas (Merz and Mosley, 1998). Although water discharge from the scar was greater than from other areas, P sources there were not significant because soil is very thin or nonexistent and/or livestock do not congregate in that area. The scar is too steep and does not have enough grass for the animals. Other unmonitored source areas where livestock waste accumulates or fertiliser is applied contribute P to the stream. Enhanced erosion of debris and transport of unsampled particulate P can also result in elevated DRP in streamwater as the particulate P dissolves over time (McColl, 1978; Sharpley *et al.*, 1994; Daniel *et al.*, 1994; Merz and Mosley, 1998). However, the colluvium and undisturbed areas that we monitored appear to contribute some P to the stream.

Total Dissolved Solids

Total dissolved solids were high in surface runoff from the scar (up to 412 mg L^{-1}) during the storm of 9 April 1998 (Table 2). Values also were elevated in runoff from colluvium and undisturbed areas during that storm (up to 38 mg L^{-1}) compared to other storms (1-4 mg L^{-1}). However, dissolved solids were lower in runoff from debris during the 9 April storm (36-57 mg L^{-1}) relative to other storms (136-217 mg L^{-1}). In subsurface water, values were much more consistent across CLC classes (214-280 mg L^{-1}), with the exception of subsurface water in colluvium on 9 April (10 mg L^{-1}). Values in soil water were also consistent (230-320 mg L^{-1}), except at one of the undisturbed sites (40-53 mg L^{-1}). Dissolved solids concentrations were relatively consistent in the stream during the storm of 22 August (Fig. 4c). Although a cation-anion balance was not done or used to confirm the dissolved solids values, the relatively high values were consistent with the associated elevated concentrations of the few key ions that were analysed, including calcium, magnesium, chloride and sulfate.

Subsurface and soil water had higher median concentrations than the stream, and subsurface water had a higher value than surface runoff (Table 2, Fig. 8a). The mean concentration in subsurface water was $\geq$20% more than the value in the stream ($P<0.05$). For a difference of 10% or $P<0.1$, soil water also had a higher value than the stream.

There were no clear patterns in concentrations of dissolved solids in water at different depths (types of water) in debris on 17 August and 6 September (Fig. 6c). This could result from differences in the timing of chemical movement downward from the upper soil layers. Values were consistently elevated in shallow soil water and saturated subsurface water.

Subsurface and soil water had higher total dissolved solids than stream water and is a source of solutes and a pathway to the stream. Subsurface water also had somewhat higher values than surface runoff. This is expected, given the longer residence time of water in the saturated and unsaturated zones, relative to surface runoff and the stream, and chemical leaching in

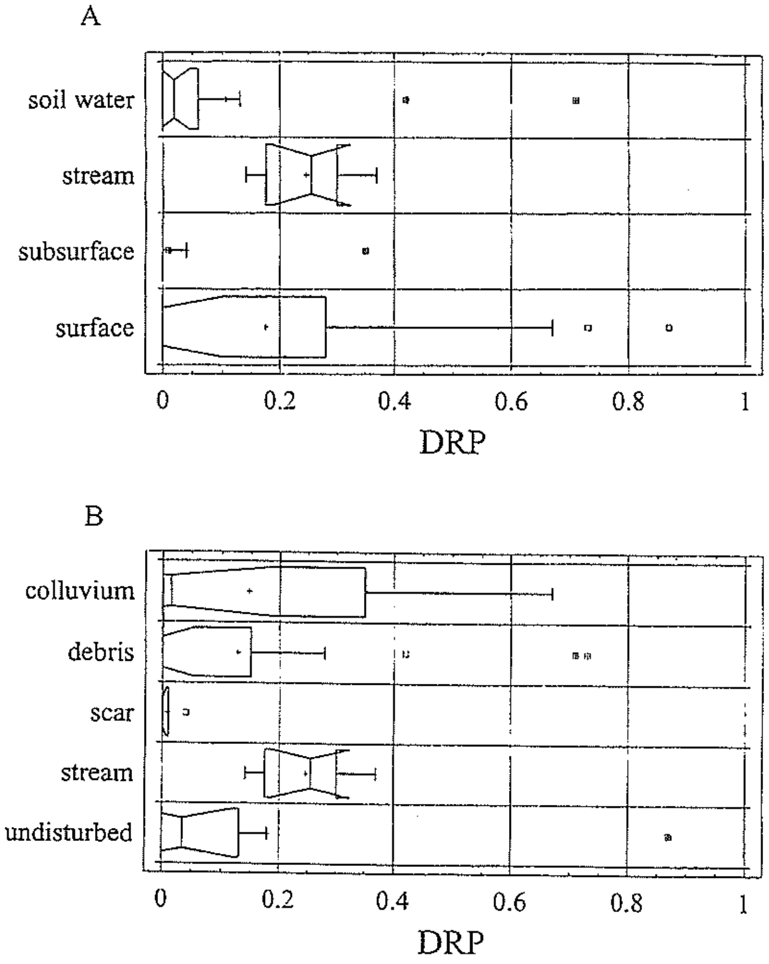

Figure 7 – Boxplots of DRP concentrations (mg L^{-1}) by (A) types of water and (B) CLC class.

the soil matrix (Pionke *et al.*, 1988). Subsurface values were slightly higher than concentrations in soil.

The scar had higher ($P<0.05$) total dissolved solids than stream water and the undisturbed area, and debris had a higher value than colluvium, the stream and the undisturbed area (Table 3, Fig. 8b). Dissolved solids in the stream were also higher than in the undisturbed area. The concentration in debris was ≥20% more than in colluvium, the undisturbed area and the stream, and the dissolved solids value for the scar was ≥20% more than in the undisturbed area and stream ($P<0.05$). For $P<0.1$ the value in the scar was also ≥20% more than in colluvium.

This analysis shows that total dissolved solids were higher in areas recently disturbed by landslides than in other CLC classes and in the stream. Values in stream water were also somewhat higher than in the undisturbed area. The landslide-disturbed areas, therefore, were sources of solutes transported to the stream via subsurface pathways. However, these results differ from those for NO_3^--N, which was much higher in colluvium and undisturbed areas than in disturbed areas. DRP was also highest in the stream, with elevated values in colluvium. With the exception of nutrients, therefore, many solutes were derived from disturbed areas, and the undisturbed areas were contributing fewer solutes but a greater proportion of NO_3^--N and DRP.

Discharge

In surface runoff during the storm of 21-22 August 1997, specific discharges ranged from 0.02 L $m^{-2}s^{-1}$ (2.1 L s^{-1}) from one of the colluvium sites to 1.2 L $m^{-2}s^{-1}$ (210 L s^{-1}) from debris (Table 4). Specific volumes of water were greatest from debris (14.8 L m^{-2} or 2550 L), and lowest from one of the undisturbed sites (0.08 L m^{-2} or 3.4 L. Therefore, debris generated substantially more runoff than undisturbed areas and colluvium. The total length of time for discharge ranged from 149 minutes for one of the undisturbed sites to 752 minutes for one of the colluvium sites. Surface runoff in this catchment can be generated from both Hortonian and saturation excess overland flow, primarily from partial and variable source areas in near-stream riparian areas, valley bottoms, and swales (Merz and Mosley, 1998)

Subsurface discharge was not as variable across sites. Values were greatest for colluvium (1.7 L $m^{-2}s^{-1}$ or 180 L s^{-1}) and lowest from one of the debris sites (0.2 L $m^{-2}s^{-1}$ or 64 L s^{-1}). Specific volumes ranged from 3.6 L m^{-2} (980 L) for one of the debris sites to 5.8 L m^{-2} (620 L) for colluvium. The length of time for discharge ranged from 206 minutes for colluvium to 927 minutes for the debris site with the larger flow.

The greater subsurface discharge from colluvium could be a function of the more structured nature of the colluvial material relative to the mixed

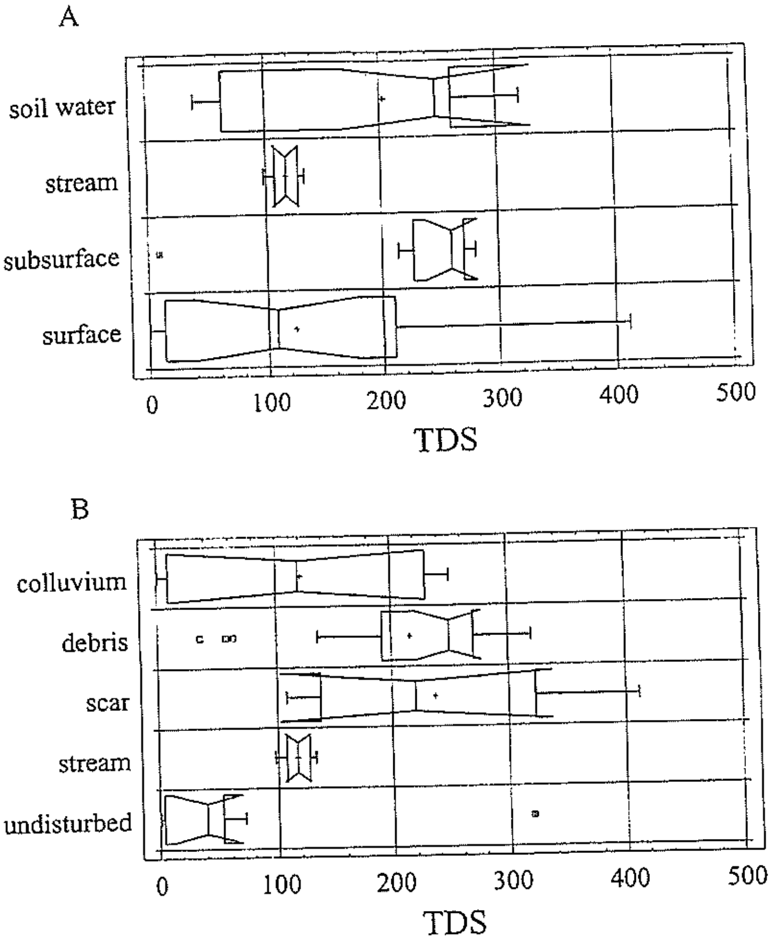

Figure 8 – Boxplots of total dissolved solids concentrations (mg L^{-1}) by (A) types of water and (B) CLC class.

nature of debris. The undisturbed colluvium could allow greater matrix flow. Matrix, macropore, and pipe flow can all form part of subsurface flow, and in combination can produce stream flow responses as rapid as surface runoff and contribute a large portion of the observed streamflow (Mosley, 1979, 1982; Pearce *et al.*, 1986; Sklash *et al.*, 1986; Anderson and Burt, 1990). Recent studies in New Zealand and overseas also indicate that most water reaching channels is displaced old subsurface water (Stewart and Rowe, 1994; Bidwell and Stewart, 1996). Much of the subsurface flow that contributes to stormflow is transported in topographic hollows discharging to steep, first–order ephemeral channels, such as are characteristic of this catchment. The flow can be controlled by the bedrock topography in saturated or easily saturated areas, where old water is stored in the regolith (Anderson and Burt, 1978, 1990; McDonnell *et al.*, 1996).

The locations and volumes of subsurface flow could also vary significantly across the hillslope and change over time (Woods and Rowe, 1996), and in some catchments appear to be largely controlled by topographic convergence and divergence (Woods and Rowe, 1996). This variability cannot be adequately quantified using the sampling regime in this study and could pose a problem if discharge values were extrapolated across the hillslope to estimate total values for the catchment. However, this type of extrapolation is not being done as part of this study, and the locations sampled are considered representative of the CLC classes within this small catchment.

Loads

NO_3^--N

Although surface runoff from debris was very large during the 21-22 August storm, NO_3^--N was not detected (Table 4). The smallest measured specific load of NO_3^--N in runoff was 0.01 mg $m^{-2}s^{-1}$ (0.4 mg s^{-1}) from one of the undisturbed areas. One of the colluvium locations had the greatest load (0.21 mg $m^{-2}s^{-1}$ or 9.1 mg s^{-1}). However, the load from the other colluvium site was less than that from the other undisturbed location. Total measured NO_3^--N loads for the storm ranged from 0.02 mg m^{-2} (1 mg) for one of the undisturbed sites to 1.1 mg m^{-2} (48 mg) for one of the colluvium sites. Again, the load from the other colluvium location was less than that from the second undisturbed site.

NO_3^--N loads from subsurface areas were greater than from surface runoff, primarily reflecting the greater subsurface discharges. They ranged from 0.06 mg $m^{-2}s^{-1}$ (16 mg s^{-1}) for one of the debris sites to 0.51 mg $m^{-2}s^{-1}$ (54 mg s^{-1}) for colluvium. Total loads were also greatest from colluvium (1.8 mg m^{-2} or 190 mg), reflecting both the higher concentrations in and greater discharges from colluvium.

Table 4 – Discharges, concentrations and loads for storm of 22 August 1997

Location	CLC Class	Area (m^{-2})	Discharge (L)	$(L\ m^{-m2})$	(Min)	$(L\ s^{-1})$	$L\ m^{-2}s^{-1})$	N Conc $(mg\ L^{-1})$	N Load $(mg\ s^{-1})$	$(mg\ m^{-2}s^{-1})$	(mg)	$(mg\ m^{-2})$
Surface	Colluvium	111	27	0.20	752	2.1	0.02	0.60	1.3	0.01	16	0.15
Surface	Colluvium	44	68	1.50	313	12.0	0.30	0.70	9.1	0.21	48	1.10
Surface	Debris	172	2550	14.80	721	210	1.20	0.00	0.0	0.00	0	0.00
Surface	Undisturbed	43	18	0.40	741	1.4	0.03	0.50	0.7	0.02	9	0.20
Surface	Undisturbed	45	3.4	0.08	139	1.4	0.03	0.30	0.4	0.01	1	0.02
Subsurface	Colluvium	106	620	5.80	206	181	1.70	0.30	54	0.51	190	1.80
Subsurface	Debris	271	980	3.60	927	64	0.20	0.25	16	0.05	250	0.90
Subsurface	Debris	228	1160	5.10	618	112	0.50	0.15	17	0.07	170	0.80

Location	CLC Class	Area (m^{-2})	TP Conc $(mg\ L^{-1})$	TP Load $(mg\ L^{-1})$	$(mg\ m^{-2}s^{-1})$	(mg)	$(mg\ m^{-2})$	N Conc $(mg\ L^{-1})$	TDS Load $(mg\ s^{-1})$	$(mg\ m^{-2}s^{-1})$	(mg)	$(mg\ m^{-2})$
Surface	Colluvium	111	0.062	0.13	0.001	1.7	0.02	2	4.3	0.04	54	0.5
Surface	Colluvium	44	0.056	0.73	3.8	0.09	0.09	1	13.0	0.30	68	1.5
Surface	Debris	172	0.090	19	0.111	230	1.30	164	34700	200	417500	2430
Surface	Undisturbed	43	0.036	0.05	0.001	0.6	0.02	1	1.4	0.03	18	0.4
Surface	Undisturbed	45	0.152	0.21	0.005	0.5	0.01	4	5.5	0.12	14	0.3
Subsurface	Colluvium	106	ND	ND	ND	ND		214	38500	360	132300	1250
Subsurface	Debris	271	ND	ND	ND	ND		253	16100	59	248700	920
Subsurface	Debris	228	ND	ND	ND	ND		251	28200	120	290200	1270

ND - no data collected CLC - contemporary land-surface condition TP - total P TDS - total dissolved solids

Phosphorus

DRP loads in surface runoff and subsurface flows during the August storm were not significant, because it was usually not detected at the sites measured (Table 4). The only measured load was in surface runoff from debris (<0.0001 mg $m^{-2}s^{-1}$ or <2 mg m^{-2}). However, total P loads in runoff during that storm were considerably greater than DRP (Table 4, total P was not analysed in subsurface water). Particulate P contributes to most of the total P, and erosion and transport of particulate P can be significant source of P to streams in this catchment (McColl, 1978). The greatest specific total P load was from debris (0.11 mg $m^{-2}s^{-1}$ or 19 mg s^{-1}) and the smallest was from undisturbed areas (0.001 and 0.005 mg $m^{-2}s^{-1}$). Loads from colluvium were also low. However, the total P concentration in one of the undisturbed areas (0.152 mg L^{-1}) was higher than in debris (0.09 mg L^{-1}), possibly from a waste source in the undisturbed area. As discussed earlier, DRP concentrations were also very low in water from debris. The high total P load from debris, therefore, was primarily a result of the greater discharge and erosion of debris material with associated particulate P. Total P loads in surface runoff for the storm ranged from 0.01 mg m^{-2} (0.5 mg) from the undisturbed area to 1.3 mg m^{-2} (230 mg) from debris. This correlates well with the only measured DRP loading, which was from debris.

Total Dissolved Solids

Specific total dissolved solids loads in surface runoff were lowest from one of the undisturbed areas (0.03 mg $m^{-2}s^{-1}$ and 1.4 mg s^{-1}, Table 4); they were also low from colluvium. Loadings were much greater from debris (200 mg $m^{-2}s^{-1}$ and 34700 mg s^{-1}). Total loads were also extremely large from debris (2430 mg m^{-2} and 418 g). Specific loads in subsurface water were similar to those in runoff from debris. They were lowest from one of the debris sites (59 mg $m^{-2}s^{-1}$ or 16100 mg s^{-1}) and highest from colluvium (360 mg $m^{-2}s^{-1}$ or 38500 mg s^{-1}). However, the highest total loads were from the other debris site (1270 mg m^{-2} and 290 g), due to its larger total flow and drainage area.

NO_3^--N was not detected and, therefore, did not contribute to the total dissolved solids load in surface runoff from debris, but some DRP was part of the load. The pattern of dissolved solids load in subsurface water was similar to that for the NO_3^--N load: specific loads were greatest from colluvium, but total loads were highest from debris because of the greater total flows.

Summary and conclusions

In steep, pastoral catchments in hill country severely affected by landslides, soil water in undisturbed areas can be a significant source of NO_3^--N to streams. This NO_3^--N is derived from the accumulation of animal waste in these areas, from fertiliser, and from N fixation. Concentrations can also be diluted in streams, taken up by aquatic biota, and transported rapidly downstream. These characteristics are similar to those in catchments undisturbed by landslides. However, the physical disturbances caused by landslides alter livestock usage of areas, thereby affecting the spatial patterns of NO_3^--N sources and transport from those sources to streams.

In contrast, DRP concentrations in the stream were significantly higher than values in other types of water in the areas sampled. Unmonitored sources areas could be contributing P to the stream during the storms sampled. Although landslide-disturbed areas are not significant sources of DRP, they contribute total P (primarily as particulate P) through enhanced erosion and transport of surface material that can be a significant source of elevated DRP values in streams. As with NO_3-N, the spatial patterns of P sources and transport from sources to streams are altered in landslide areas due to changes in patterns of livestock grazing.

Results for total dissolved solids showed that, with the exception of dissolved nutrients, debris can be a significant source of solutes transported to streams via subsurface pathways. Undisturbed areas generally do not contribute many solutes besides NO_3^--N.

This study has demonstrated that landslides can alter the sources and transport of contaminants in steep, pastoral hill country. Not only does physical disturbance directly influence contaminant sources and transport (particularly of P), it indirectly affects them by altering livestock usage of areas. This indirect effect may be more significant, in terms of waste sources and N, than the direct influence of physical disturbance. These effects should be considered when developing contaminant transport and catchment water quality models for landslide-disturbed areas. Models should be able to accommodate changes in N and P source areas due to physical impacts of the landslides and their effects on livestock behaviour. Landslides also have management implications regarding downstream receiving water quality problems, and targeting best management practices. Water quality problems due to particulate P, including dissolution of P and eutrophication over time, can be worse in catchments with landslides. Management measures in such areas should focus on remediation of landslide areas, sediment control, and management of livestock movement to reduce erosion induced by animals, and accumulation and leaching of nutrients from areas where animals congregate.

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